Nature Conservation

Nature Conservation 54: 179–202 (2023)
DOI: 10.3897/natureconservation.54.110257

Research Article

Vegetation changes along an urbanisation and atmospheric pollution gradient in Mexico

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Abstract

Green areas are important places for biodiversity conservation within cities, but their vegetation is affected by various anthropogenic factors. This study used an exploratory approach to examine the influence of urbanisation and air pollution-related factors on the indicators for the composition and structure of vegetation in an urban area in northeast Mexico. Based on the spatial analysis of the major air pollutants, four sampling categories were delimited (rural, low, moderate and high urbanisation). The differences between categories, based on vegetation structure, were determined using non-parametric Kruskal-Wallis tests. The Importance Value was calculated for the species. The floristic similarity was compared using NMDS and PERMANOVA unidirectional. The relationship between environmental variables and abundance of species was evaluated using CCA. One hundred and ten plant species were collected, including ten alien species. The highest abundance and species richness were registered in the rural site. The general tendency of vegetation structure is to plants decreasing with respect to the increase in the levels of urbanisation and air pollution present in the study area. The association between the environmental variables and plant communities along the urbanisation gradient was significant, being the relative humidity, the particles lower than 2.5 µm, the dew point and the heat index as the most important variables. The understanding of the nature and variability of vegetation within green areas contributes to increasing our knowledge about the distribution of the environmental services they provide and the composition of the faunal communities that depend on them. For this reason, this study relates the plants of a specific area of northeast Mexico with the environmental quality present in an urban area.

Key words: air pollution, environmental variability, Monterrey Metropolitan Area, urbanisation, vegetation structure

OPEN ACCESS

Academic editor: Cássio Cardoso Pereira
Received: 29 July 2023
Accepted: 25 November 2023
Published: 21 December 2023

ZooBank: https://zoobank.org/A5AB18E6-E50C-4932-8084-76453E584C7A

Citation: Meléndez-Jaramillo E, Sánchez-Castillo L, Segura Martínez MaTJ, Sánchez-Reyes UJ (2023) Vegetation changes along an urbanisation and atmospheric pollution gradient in Mexico. Nature Conservation 54: 179–202. https://doi.org/10.3897/natureconservation.54.110257

Introduction

The demographic growth dynamics faced by cities represent a serious threat to the environment, as well as to the health and quality of life of its inhabitants (Vlahov and Galea 2002). The unsustainable use of natural resources, intense land-use changes, increasing density of urban/industrial centres and the

growing emission of pollutants irreversibly damage the environment (García et al. 2013). These effects not only harm living beings, but also generate phenomena that affect the ecosystem (López et al. 2001). Likewise, the accelerated urbanisation changes the structure of cities and affects their climate and that of their surrounding area (Tang et al. 2008). This urbanisation process occurs more rapidly in countries located in regions classified as developing economies. Particularly in Latin America, where it is estimated that 75% of the population live in cities (UN-HABITAT 2010).

In Mexico, air pollution has deteriorated air quality in various cities, including the Metropolitan Area of the Valley of Mexico, the Metropolitan Area of Guadalajara and the Monterrey Metropolitan Area (MMA) (García et al. 2012; Cerón et al. 2014; Mancilla et al. 2015; Menchaca et al. 2015). It is appropriate to point out that there is a perception problem in society as there is no clear awareness of pollutant emissions, their concentrations and damage to health, urban infrastructure and ecosystems (Lezama and Graizbord 2010). The State of Nuevo León, in the northeast of Mexico, has an unregulated urban growth. Its main urban sprawl, the MMA presents serious environmental problems: geological and hydrological risks, water scarcity, loss of green areas, air pollution, amongst many others (Badillo et al. 2015; Orta et al. 2016; Sanchez-Castillo et al. 2016; Sisto et al. 2016; Ybáñez and Barboza 2017).

Studies of species diversity in urban ecosystems are needed to understand the effect of anthropogenic development on ecosystem integrity and sustenance (Mukherjee et al. 2015). To study the effects of urbanisation on ecosystem structure and function, researchers have used the urban-rural gradient methodology (Pennington et al. 2010). Urban-rural gradients are generally realised on large spatial scales and, in some cases, have been conceived as a linear transect radiating from the city centre towards less disturbed landscapes. Studies employing this method have documented declines in plant species diversity, basal area and density of native species as sites become more urbanised. These studies, which show a decrease in species richness as urbanisation increases, follow a general disturbance hypothesis (Porter et al. 2001; Moffatt et al. 2004; Burton et al. 2005; Duguay et al. 2007).

On the other hand, the intermediate disturbance hypothesis has been one of the main models used to interpret urban plant diversity patterns (Johnson and Swan 2014). The theory has been applied to explore the co-existence of native and non-native species along urban-rural gradients or within the urban environment between patches that vary in level of disturbance (e.g. Porter et al. (2001); ManSecak and Wein (2006); Catford et al. (2012)). The expectation is that species diversity will be maximised in intermediate locations, where native and invasive species are found in the same communities, in relatively uniform proportions.

Previous studies of large-scale urban-rural gradients have documented that those urban forests are more deteriorated than their “natural” or rural counterparts (Paul and Meyer 2001; Güler 2020). Consequently, they reduce the perceived ecological value of remnant vegetation within highly modified landscapes. However, it is important to understand the potential ecological and social value of remnant urban vegetation (Turner et al. 2004; Czaja et al. 2020). Given that more than 60% of the world’s population will reside in urban areas by 2050, these forest fragments in urban settings could provide critical ecosystem

services for both people and other species (Bernhardt and Palmer 2007; Zegeye et al. 2023).

For our study, we characterised the remnant vegetation of the MMA, northeast Mexico, along an urbanisation gradient, based on parameters of atmospheric pollution. The objectives of this study were: (1) Identify the plant species richness in the MMA, northeast Mexico; (2) Compare the variation in richness, abundance and diversity of plant species amongst urbanisation categories; (3) Quantify the value of importance of the species by urbanisation category; and (4) Analyse the influence of environmental variation (air pollutants, climatic factors and soil) on the abundance and richness of plant species. Our hypothesis is that the structure and composition of the vegetation decrease with respect to the increase in urbanisation levels in the MMA.

Methods

Study area

The MMA is the largest urban area in northeast Mexico and the third largest urban centre in the country, extending from 25°15' to 26°30' north latitude and from 99°40' to 101°10' west longitude (Fig. 1A, B). The area is bounded by the coastal plain of the Gulf of Mexico and the Sierra Madre Oriental Mountain Range. Several municipalities compose the geographical area of MMA: Apodaca, Cadereyta, García, General Escobedo, Guadalupe, Jiménez, Juárez, Monterrey, Salinas Victoria, San Nicolás de los Garza, San Pedro Garza García, Santa Catarina and Santiago (Alanís 2005; González et al. 2011; Mancilla et al. 2015). The main vegetation cover found at MMA is forest, scrubs and grasslands (Carpio et al. 2021). The MMA has a vehicle fleet of 2.5 million vehicles (Castillo-Nava et al. 2020) and 5.3 million inhabitants (INEGI 2021), which is probably even higher today. Likewise, there is a variety of industrial complexes that include the production of glass, steel, cement and paper, amongst others (Menchaca et al. 2015). The city centre has an average altitude of 540 m a.s.l., the characteristic climate is dry steppe, hot and extreme with temperatures above 35 °C during the summer and below 8 °C during the winter (Alanís 2005; González et al. 2011; Menchaca et al. 2015).

Delimitation of the urbanisation gradient

Since November 1992, the MMA has operated a network of air quality monitoring stations known as the Integral Environmental Monitoring System (SIMA). The SIMA network is currently made up of 14 recording stations distributed according to criteria from meteorological, land use and population density studies. The measurements recorded at these monitoring stations are: PM_{10} (particulate matter less than 10 μm), $PM_{2.5}$ (particulate matter less than 2.5 μm), carbon monoxide (CO), ozone (O_3), nitrogen oxides (NO_x) and sulphur dioxide (SO_2). In addition, some meteorological variables are reported, such as barometric pressure, rainfall, relative humidity, solar radiation, temperature and wind direction and magnitude (Arreola and González 1999; González et al. 2011; Mancilla et al. 2015). The data recorded by the SIMA stations for air quality and meteorological variables (2009–2018) were obtained from the National

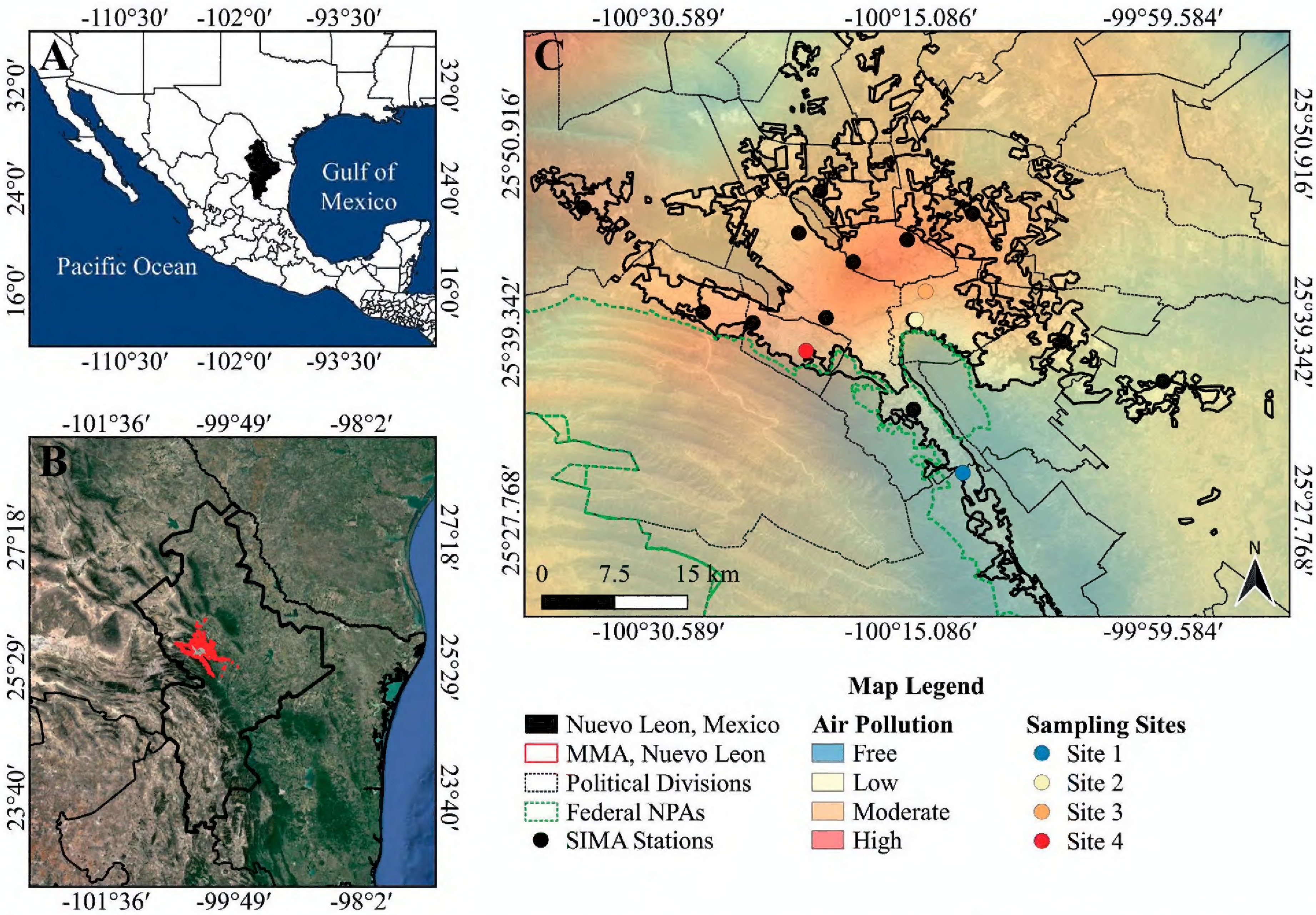

Figure 1. Study area and location of sampling sites **A** location of Nuevo Leon in Mexico **B** location of the MMA inside Nuevo Leon **C** location of sampling sites according with the air pollution levels.

Air Quality Information System (SINAICA). Obtaining descriptive measures for each year and for each of the recording stations was carried out in the Statistica 13.3 programme (TIBCO Software Inc. 2017).

To identify the main pollutants that describe air quality in the MMA during the 2009–2018 period, a Principal Component Analysis (PCA) was carried out. Subsequently, to differentiate the changes in the spatial distribution of pollutants that are indicators of air quality in the MMA, maps were created using the annual average information on each monitoring station. Mapping was done using Inverse Distance Weighting (IDW) interpolation, with a Distance Coefficient of 2 and the output raster pixel size reset to 15 metres. As a reference of the extension, the minimum and maximum distances of the vector sections corresponding to the urban areas that make up the MMA were taken; these sections were obtained from the national layer of Land Use and Vegetation Series 6 (INEGI 2016). The procedures described above were performed using Quantum GIS 3.2 software (Quantum GIS Development Team 2018). As a result, four categories of urbanisation by atmospheric pollution were generated: rural (lower than 3.22 µg/m^3 of $PM_{2.5}$), low (3.22 to 10.56 µg/m^3 of $PM_{2.5}$), moderate (10.56 to 17.92 µg/m^3 of $PM_{2.5}$) and high (17.92 to 25.3 µg/m^3 of $PM_{2.5}$) (Fig. 1C).

Selection of sampling sites

Four permanent sampling sites were delimited, based on the spatial superposition of three geographic elements: (1) the interpolation of the main air pollutants

was used for determination of the urbanisation gradient in the study area (Fig. 1C); (2) images obtained from the Google Earth Pro software were used to differentiate the spatial presence or absence of vegetation cover and (3) a mesh with a grid size of 150 × 150 m was delimited to select sampling areas with complete vegetation cover. Overlay and selection procedures were performed in Quantum GIS 3.2 software. The rural site is located in the Municipality of Santiago, a rural area without substantial urbanisation or air pollution and with secondary submontane scrub vegetation (25°30'41.184"N, 100°11'53.159"W). The low urbanisation site is located in the central zone of the Municipality of Guadalupe with low values of air pollution and secondary vegetation of submontane scrub (25°40'4.944"N, 100°14'45.564"W). The moderate urbanisation site is located in the northern zone of the Municipality of Guadalupe with moderate air pollution and secondary vegetation of submontane scrub (25°42'44.017"N, 100°13'58.825"W). The high urbanisation site is in the Municipality of San Pedro Garza García with high air pollution and anthropogenic submontane scrub vegetation (25°38'11.112"N, 100°21'30.815"W) (Fig. 1C).

Sample collection and processing

During April 2019, the analysis of preliminary samples obtained in the study area was carried out. The Clench model was used to calculate the minimum sample size to be used, based on the method and parameters indicated by Jiménez-Valverde and Hortal (2003). According to the analysis, between 5 to 8 sampling units are needed to register the 95% of the richness of each site. Vegetation was assessed using 20 quadrats of 10 × 10 m (0.01 hectares), which were evenly distributed amongst the four categories of urbanisation by air pollution (five quadrats per category). The quadrats were located from inside the sampling site (150 × 150 m) (2.25 hectares) randomly and they were placed using the tool random points inside a polygon in Quantum GIS 3.2 software. The quadrats are located in patches of natural and native vegetation. The evaluation was carried once per season: dry season (November to April) and rainy season (May to October), during the period from May 2019 to April 2020. The seasons were defined, based on the historical data of the monthly total values of temperature and rainfall (average from 2009 to 2018), which were obtained from the SIMA stations located within the study area (Fig. 2).

Measurements were carried out independently for each of the vegetation strata. For the herbaceous stratum, five sub-quadrats of 1 × 1 m (5 m^2 in total per quadrant) were delimited. In the shrub stratum, two 5 × 5 m sub-quadrats (50 m^2 in total per quadrant) were evaluated. Finally, the tree stratum was evaluated in the entire quadrat, 10 × 10 m (100 m^2 in total per quadrat). The dimension of the quadrat and sub-quadrats was established according to the criteria described by Brower et al. (1998).

In each quadrat/sub-quadrats, the following measurements were made: (1) height of the plant (from its base at ground level to the highest branch); (2) largest and (3) smallest diameter of the aerial projection of the plant. The number of individuals of each morphospecies assigned in the field was quantified and their identification in the laboratory was carried out using the works of Alanís and González (2003), Stubbendieck et al. (2003) and Zurita and Elizondo (2009); likewise, the botanical nomenclature was homogenised using

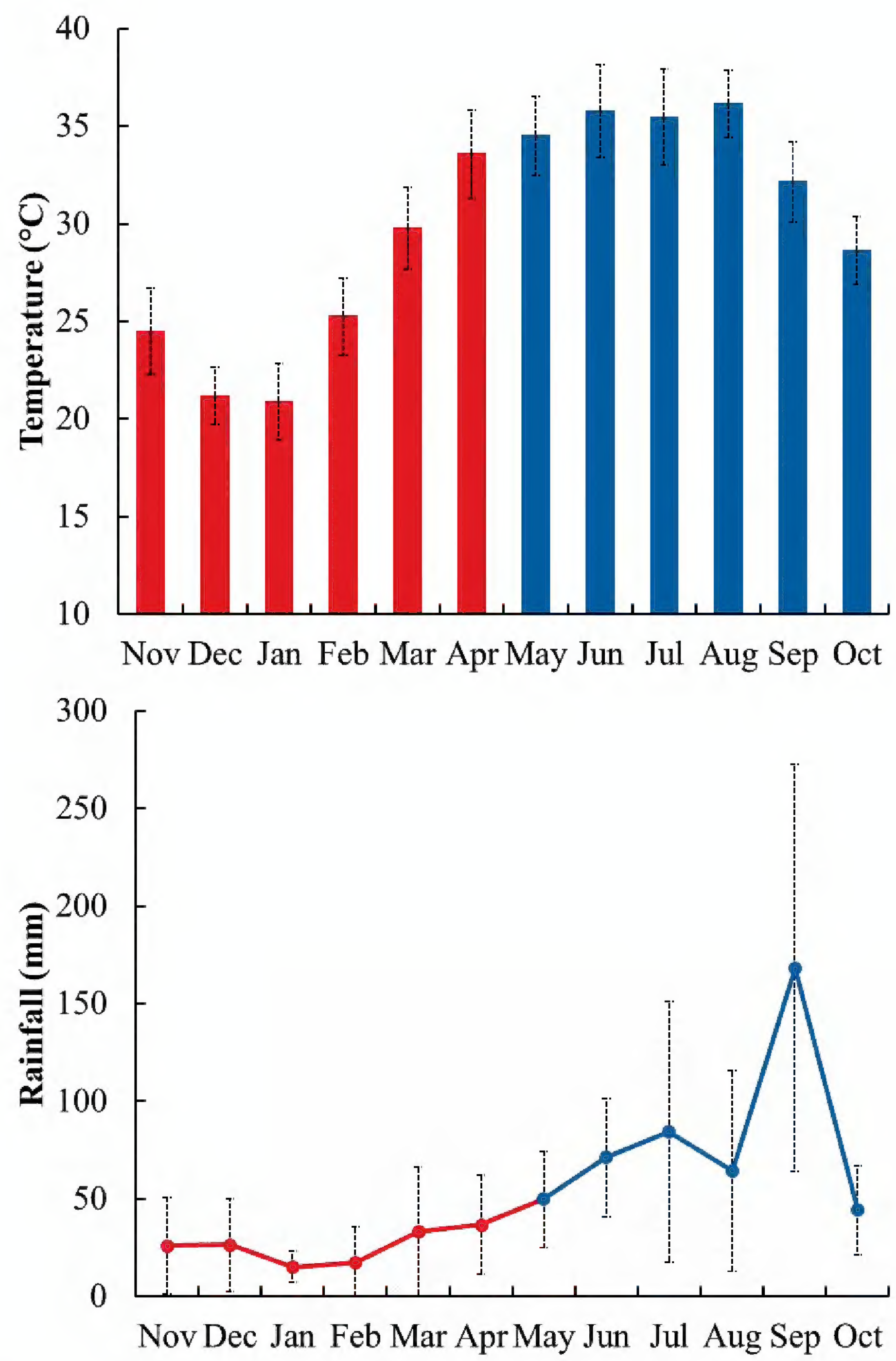

Figure 2. Monthly average variation of temperature and rainfall in the MMA. Dry season (red colour) and Rainy season (blue colour).

the International Plant Names Index base (IPNI 2022). Villaseñor and Espinosa-García (2004) were mainly followed to determine which plant species were not native to the MMA region.

Microenvironment measurement

The microenvironmental variables were measured in each of the quadrants using a Kestrel 5500 portable weather station, a CEM – DT1308 digital luxmeter, a CEM – DT9881 particle counter and a HB – 2 soil moisture and pH meter, simultaneously with the sampling of the vegetation, recording the following variables: maximum wind speed (MWS) and average wind speed (AWS) (obtained during five minutes of exposure), temperature (T), relative humidity (RH), heat index (HI), dew point (DP), evapotranspiration (E), solar radiation (SR), particles of 2.5 ($PM_{2.5}$) and 10 microns (PM_{10}), soil pH (SpH) and soil moisture (SM). Measurements were carried out in the centre of each quadrat during the early hours of the morning, noon and before sunset, avoiding direct solar radiation.

Data analysis

Species richness was measured as the total number of species observed in the study area, as well as in each of the sites. Significant differences in the

number of species between sites were determined using non-parametric Kruskal-Wallis tests, in Statistica 13.3 software. Sampling efficiency was calculated for the entire study area and for each site using the interpolation and extrapolation methodology proposed by Chao and Jost (2012), available in the iNEXT package (Hsieh et al. 2016) for version 3.5.3 of R (R Development Core Team 2019).

Differences in plant abundance between sites were calculated with a Kruskal-Wallis test. For the analysis of alpha diversity, we adopted the analytical method of Chao and Jost (2015) to obtain profiles in which diversity is evaluated in terms of "effective numbers of species" (qD), an approach that is equivalent to the numbers of Hill (Hill 1973). Hill numbers include three widely-used measures as special cases: species richness ($q = 0$), Shannon diversity (the exponential of Shannon entropy, $q = 1$) and Simpson diversity (the inverse of Simpson concentration, $q = 2$), all of which are expressed in units of "species equivalents". The analysis was performed for the entire study area and for each site using the SpadeR package (Chao et al. 2016), in R 3.5.3.

Vegetation cover was calculated according to the criteria described by Ramírez (2006). Differences in vegetation cover between sites were determined with a Kruskal-Wallis test. To examine differences in species composition between sites, we performed non-metric multidimensional scaling (NMDS) analysis, using the Bray-Curtis Index as the similarity matrix. A PERMANOVA was also performed to test for differences in species composition between sites. Both analyses were performed using the Vegan package (Oksanen et al. 2019) in R 3.5.3.

For each species, its abundance was determined according to the number of individuals, its dominance based on cover and its frequency based on its presence in the sampling quadrats. These results were used to obtain a weighted value at the taxon level called Importance Value (IV), which acquires percentage values on a scale from 0 to 100 (Mueller-Dombois and Ellenberg 1974). The IV was calculated for each site separately.

Finally, a canonical correspondence analysis (CCA) was carried out to determine the relationship between the microenvironmental variables and the abundance of the recorded species in each plot, which also includes a Monte Carlo permutation test to evaluate the significance of the microenvironmental variables in the analysis. For the CCA, the average values of the microenvironmental variables of each season of the year were used (dry and rainy season). The CCA was done using the Vegan package in R 3.5.3.

Results

A total of 12,878 plants of 42 families, 104 genera and 110 species were quantified. From this total, 17 species (594 individuals) were trees, 34 (2,595 individuals) were shrubs and 59 (9,689 individuals) were herbaceous (Table 1). The greatest abundance and richness of tree species in the study area was found in the Fabaceae family with 35.0 and 23.5% of the total registered, respectively. Likewise, Fabaceae presented the highest abundance and richness of shrub species with 29.3 and 29.4% of the total registered, respectively. Asteraceae showed the highest abundance and richness of herbaceous species with 21.6 and 18.6% of the total recorded, respectively (Table 1).

Table 1. Taxonomic list, abundance and IV of the species found in an air pollution gradient in the MMA. Legend: Site 1 = Rural, Site 2 = Low urbanisation, Site 3 = Moderate urbanisation, Site 4 = High urbanisation.

Taxon	Key	Abundance				IV			
		Site 1	Site 2	Site 3	Site 4	Site 1	Site 2	Site 3	Site 4
Tree									
Boraginaceae Juss.									
Ehretia anacua (Terán & Berland.) I.M. Johnst.	Eana	16	6	0	0	8.1	4.7	0.0	0.0
Cannabaceae Martinov									
Celtis laevigata Willd.	Clae	26	22	0	0	12.1	12.7	0.0	0.0
Ebenaceae Gürke									
Diospyros texana Scheele	Dtex	22	0	0	0	10.7	0.0	0.0	0.0
Fabaceae Lindl.									
Ebenopsis ebano (Berland.) Barneby & J.W. Grimes	Eeba	20	28	16	0	10.2	14.6	17.3	0.0
Havardia pallens (Benth.) Britton & Rose	Hpal	18	12	6	0	9.6	8.7	5.7	0.0
Leucaena leucocephala (Lam.) de Wit*	Lleu	0	26	16	0	0.0	12.6	18.9	0.0
Prosopis glandulosa Torr.	Pgla	26	18	22	0	12.4	9.6	21.2	0.0
Fagaceae Dumort.									
Quercus fusiformis Small	Qfus	0	0	0	24	0.0	0.0	0.0	22.4
Juglandaceae DC. ex Perleb									
Carya illinoinensis (Wangenh.) K. Koch	Cill	0	0	0	6	0.0	0.0	0.0	5.1
Oleaceae Hoffmanns. & Link									
Fraxinus americana L.*	Fame	0	0	14	16	0.0	0.0	16.0	16.9
Ligustrum lucidum W.T. Aiton*	Lluc	0	0	0	22	0.0	0.0	0.0	18.0
Rutaceae Juss.									
Sargentia greggii S. Watson	Sgre	20	0	0	0	9.3	0.0	0.0	0.0
Salicaceae Mirb.									
Salix nigra Marshall	Snig	0	22	0	0	0.0	11.8	0.0	0.0
Sapindaceae Juss.									
Koelreuteria elegans (Seem.) A.C. Sm.*	Kele	0	30	22	16	0.0	14.1	21.0	14.1
Sapindus saponaria L.	Ssap	34	0	0	0	15.4	0.0	0.0	0.0
Ungnadia speciosa Endl.	Uspe	0	0	0	26	0.0	0.0	0.0	23.5
Sapotaceae Juss.									
Sideroxylon celastrinum (Kunth) T.D. Penn.	Scel	24	18	0	0	12.2	11.2	0.0	0.0
Shrub									
Asparagaceae Juss.									
Yucca treculeana Carrière	Ytre	33	0	0	0	2.9	0.0	0.0	0.0
Asteraceae Bercht. & J. Presl									
Gochnatia hypoleuca (DC.) A. Gray	Ghyp	34	0	0	0	3.0	0.0	0.0	0.0
Bignoniaceae Juss.									
Tecoma stans (L.) Juss. ex Kunth	Tsta	0	59	0	0	0.0	7.5	0.0	0.0
Boraginaceae Juss.									
Cordia boissieri A. DC.	Cboi	54	44	0	0	4.4	6.3	0.0	0.0
Cactaceae Juss.									
Opuntia engelmannii Salm-Dyck ex Engelm.	Oeng	40	0	0	0	3.6	0.0	0.0	0.0

Taxon	Key	Abundance				IV			
		Site 1	Site 2	Site 3	Site 4	Site 1	Site 2	Site 3	Site 4
Cannabaceae Martinov									
Celtis pallida Torr.	Cpal	53	32	0	0	4.2	4.9	0.0	0.0
Capparaceae Juss.									
Capparis flexuosa Vell.	Cfle	39	0	0	0	3.4	0.0	0.0	0.0
Euphorbiaceae Juss.									
Adelia vaseyi (JM Coult.) Pax y K. Hoffm.	Avas	43	36	0	0	3.7	5.4	0.0	0.0
Fabaceae Lindl.									
Acacia berlandieri Benth.	Aber	59	40	0	0	4.7	5.9	0.0	0.0
Acacia farnesiana (L.) Willd.	Afar	0	0	53	0	0.0	0.0	12.8	0.0
Acacia rigidula Benth.	Arig	49	0	0	0	4.1	0.0	0.0	0.0
Bauhinia mexicana Vogel	Bmex	36	0	0	0	3.4	0.0	0.0	0.0
Caesalpinia mexicana A. Gray	Cmex	0	33	50	61	0.0	5.0	12.6	19.6
Dalea scandens (Mill.) R.T. Clausen	Dsca	45	0	0	0	3.9	0.0	0.0	0.0
Erythrina herbacea L.	Eher	43	0	0	0	3.8	0.0	0.0	0.0
Eysenhardtia texana Scheele	Etex	28	38	40	45	3.0	5.5	11.0	15.2
Mimosa monancistra Benth.	Mmon	0	40	43	0	0.0	5.8	11.7	0.0
Parkinsonia aculeata L.	Pacu	0	0	57	0	0.0	0.0	14.2	0.0
Lythraceae J. St.-Hil.									
Punica granatum L.*	Pgra	0	0	0	41	0.0	0.0	0.0	15.1
Malpighiaceae Juss.									
Malpighia glabra L.	Mgla	46	33	0	0	3.9	5.1	0.0	0.0
Mascagnia macroptera (Moc. & Sessé ex DC.) Nied.	Mmac	54	50	48	63	4.3	6.7	12.3	19.6
Myrtaceae Juss.									
Psidium guajava L.*	Pgua	0	0	0	41	0.0	0.0	0.0	14.6
Oleaceae Hoffmanns. & Link									
Forestiera angustifolia Torr.	Fang	29	45	0	0	2.8	6.2	0.0	0.0
Rhamnaceae Juss.									
Condalia hookeri M.C. Johnst.	Choo	49	56	0	0	4.0	7.3	0.0	0.0
Karwinskia humboldtiana (Schult.) Zucc.	Khum	51	0	0	0	4.3	0.0	0.0	0.0
Ziziphus obtusifolia (Hook. ex Torr. & A. Gray) A. Gray	Zobt	33	0	0	0	2.9	0.0	0.0	0.0
Rubiaceae Juss.									
Randia obcordata S. Watson	Robc	42	53	0	0	3.6	7.2	0.0	0.0
Rutaceae Juss.									
Helietta parvifolia (A. Gray ex Hemsl.) Benth.	Hpar	45	41	48	0	3.8	5.7	12.5	0.0
Zanthoxylum fagara (L.) Sarg.	Zfag	47	41	0	0	3.8	5.9	0.0	0.0
Salicaceae Mirb.									
Neopringlea integrifolia (Hemsl.) S. Watson	Nint	58	0	0	0	4.5	0.0	0.0	0.0
Scrophulariaceae Juss.									
Leucophyllum frutescens (Berland.) I.M. Johnst.	Lfru	35	36	0	0	2.9	5.4	0.0	0.0
Simaroubaceae DC.									
Castela erecta Turpin	Cere	49	0	0	0	3.5	0.0	0.0	0.0
Verbenaceae J. St.-Hil.									
Citharexylum berlandieri B.L. Rob.	Cber	43	0	0	0	3.2	0.0	0.0	0.0

Taxon	Key	Abundance				IV			
		Site 1	Site 2	Site 3	Site 4	Site 1	Site 2	Site 3	Site 4
Lantana camara L.	Lcam	66	26	52	47	4.5	4.1	12.9	15.9
Herb									
Acanthaceae Juss.									
Elytraria bromoides Oerst.	Ebro	79	0	0	0	3.0	0.0	0.0	0.0
Justicia pilosella (Nees) Hilsenb.	Jpil	80	114	0	0	3.0	4.2	0.0	0.0
Ruellia nudiflora (Engelm. & A. Gray) Urb.	Rnud	0	122	0	0	0.0	4.3	0.0	0.0
Tetramerium nervosum Nees	Tner	101	0	0	0	3.5	0.0	0.0	0.0
Apocynaceae Juss.									
Asclepias curassavica L.	Acur	101	76	96	90	3.5	3.2	4.0	5.5
Telosiphonia lanuginosa (M. Martens & Galeotti) Henrickson	Tlan	0	0	105	104	0.0	0.0	4.3	5.9
Asteraceae Bercht. & J. Presl									
Bidens odorata Cav.	Bodo	114	94	119	114	3.8	3.7	4.5	6.3
Calyptocarpus vialis Less.	Cvia	0	101	103	93	0.0	3.8	4.1	5.7
Chromolaena odorata (L.) R.M. King & H. Rob.	Codo	111	0	0	0	3.8	0.0	0.0	0.0
Helianthus annuus L.	Hann	0	0	94	0	0.0	0.0	3.9	0.0
Jefea lantanifolia (S. Schauer) Strother	Jlan	0	93	0	0	0.0	3.6	0.0	0.0
Sanvitalia ocymoides DC.	Socy	64	97	0	0	2.1	3.8	0.0	0.0
Simsia eurylepis S.F. Blake	Seur	0	77	95	0	0.0	3.3	4.0	0.0
Thymophylla pentachaeta (DC.) Small	Tpen	0	0	0	132	0.0	0.0	0.0	7.1
Tridax coronopifolia (Kunth) Hemsl.*	Tcor	99	101	103	131	3.4	3.8	4.2	6.8
Verbesina persicifolia DC.	Vper	61	0	0	0	2.1	0.0	0.0	0.0
Wedelia acapulcensis Kunth	Waca	96	0	0	0	3.5	0.0	0.0	0.0
Commelinaceae Mirb.									
Commelina erecta L.	Cere	99	89	95	87	3.3	3.5	3.9	5.3
Convolvulaceae Juss.									
Evolvulus alsinoides (L.) L.	Eals	0	0	119	0	0.0	0.0	4.6	0.0
Ipomoea hederacea Jacq.	Ihed	91	84	69	0	3.3	3.3	3.1	0.0
Merremia dissecta (Jacq.) Hallier f.	Mdis	0	101	98	0	0.0	3.7	3.9	0.0
Euphorbiaceae Juss.									
Acalypha monostachya Cav.	Amon	0	0	0	82	0.0	0.0	0.0	5.1
Cnidoscolus rotundifolius (Müll. Arg.) McVaugh	Crot	0	103	0	0	0.0	3.9	0.0	0.0
Croton cortesianus Kunth	Ccor	103	114	0	0	3.6	4.3	0.0	0.0
Euphorbia hirta L.	Ehir	0	78	109	0	0.0	3.3	4.3	0.0
Fabaceae Lindl.									
Canavalia villosa Benth.	Cvil	101	0	0	0	3.5	0.0	0.0	0.0
Desmanthus virgatus (L.) Willd.	Dvir	95	0	0	0	3.4	0.0	0.0	0.0
Mimosa malacophylla A. Gray	Mmal	112	0	0	0	3.7	0.0	0.0	0.0
Lamiaceae Martinov									
Ocimum micranthum Willd.*	Omic	81	76	0	0	3.1	3.2	0.0	0.0
Salvia coccinea Buc'hoz ex Etl.	Scoc	93	102	106	105	3.4	3.9	4.1	5.9
Loasaceae Juss.									
Cevallia sinuata Lag.	Csin	0	0	0	89	0.0	0.0	0.0	5.3

Taxon	Key	Abundance				IV			
		Site 1	Site 2	Site 3	Site 4	Site 1	Site 2	Site 3	Site 4
Malvaceae Juss.									
Abutilon trisulcatum (Jacq.) Urb.	Atri	0	105	113	0	0.0	3.9	4.3	0.0
Malvastrum americanum (L.) Torr.	Mame	99	0	0	0	3.4	0.0	0.0	0.0
Melochia pyramidata L.	Mpyr	78	95	99	96	2.9	3.7	4.0	5.5
Waltheria indica L.	Wind	0	0	97	0	0.0	0.0	4.0	0.0
Nyctaginaceae Juss.									
Cyphomeris crassifolia (Standl.) Standl.	Ccra	57	0	0	0	2.1	0.0	0.0	0.0
Oleaceae Hoffmanns. & Link									
Menodora heterophylla Moric. ex DC.	Mhet	0	0	63	0	0.0	0.0	2.5	0.0
Onagraceae Juss.									
Oenothera rosea L'Hér. ex Aiton	Oros	0	0	55	0	0.0	0.0	2.2	0.0
Papaveraceae Juss.									
Argemone grandiflora Sweet	Agra	0	0	63	0	0.0	0.0	2.4	0.0
Passifloraceae Juss. ex Roussel									
Passiflora foetida L.	Pfoe	69	64	0	0	2.3	2.3	0.0	0.0
Petiveriaceae C. Agardh									
Rivina humilis L.	Rhum	68	60	0	0	2.3	2.3	0.0	0.0
Poaceae Barnhart									
Aristida adscensionis L.	Aads	100	0	0	0	3.5	0.0	0.0	0.0
Bouteloua curtipendula (Michx.) Torr.	Bcur	0	0	0	104	0.0	0.0	0.0	5.9
Cenchrus spinifex Cav.	Cspi	90	114	0	0	3.3	4.2	0.0	0.0
Eragrostis barrelieri Daveau*	Ebar	0	0	84	0	0.0	0.0	3.7	0.0
Melinis repens (Willd.) Zizka*	Mrep	0	0	91	94	0.0	0.0	3.9	5.4
Panicum hallii Vasey	Phal	0	90	0	0	0.0	3.5	0.0	0.0
Paspalum unispicatum (Scribn. & Merr.) Nash	Puni	0	0	69	0	0.0	0.0	2.7	0.0
Setaria leucopila (Scribn. & Merr.) K. Schum.	Sleu	102	107	0	0	3.5	3.9	0.0	0.0
Pteridaceae E.D.M. Kirchn.									
Adiantum tricholepis Fée	Atri	92	0	0	0	3.4	0.0	0.0	0.0
Cheilanthes aemula Maxon	Caem	88	0	0	0	3.1	0.0	0.0	0.0
Ranunculaceae Juss.									
Clematis drummondii Torr. & A. Gray	Cdru	122	98	134	0	4.1	3.8	5.0	0.0
Rubiaceae Juss.									
Spermacoce glabra Michx.	Sgla	45	67	0	0	1.7	2.5	0.0	0.0
Sapindaceae Juss.									
Cardiospermum halicacabum L.	Chal	0	103	102	84	0.0	3.8	4.1	5.2
Solanaceae Juss.									
Solanum elaeagnifolium Cav.	Sela	0	0	95	0	0.0	0.0	4.0	0.0
Solanum triquetrum Cav.	Stri	54	0	0	0	1.9	0.0	0.0	0.0
Verbenaceae J. St.-Hil.									
Lantana canescens Kunth	Lcan	98	90	106	96	3.4	3.5	4.2	5.6
Phyla nodiflora (L.) Greene	Pnod	0	0	0	136	0.0	0.0	0.0	7.1
Verbena canescens Kunth	Vcan	0	0	0	112	0.0	0.0	0.0	6.4

*Introduced species.

Koelreuteria elegans (Seem.) A.C. Sm. was the species with the highest abundance of trees (68 individuals), 11.4% of the total number of individuals recorded in the study area. On the other hand, *Mascagnia macroptera* (Moc. & Sessé ex DC.) Nied. presented the highest abundance of shrubs (215 individuals) (8.3%). *Bidens odorata* Cav. was the species with the highest abundance of herbaceous plants (441 individuals) (4.6%) (Table 1). From the total of the reported species, ten are alien species, amongst which, *Koelreuteria elegans* (Seem.) A.C. Sm. and *Tridax coronopifolia* (Kunth) Hemsl. are present in most of the sampling sites (Table 1).

Variation per urbanisation category

Sapindus saponaria L. was the tree species with the highest IV (15.4%) in the rural site. On the other hand, *Ebenopsis ebano* (Berland.) Barneby & J.W. Grimes was the most important tree species (14.61%) in the low urbanisation site. *Prosopis glandulosa* Torre. was the most important species (21.21%) in the moderate urbanisation site. Likewise, *Ungnadia speciosa* Endl. was the most important species (23.48%) in the high urbanisation site (Table 1).

Acacia berlandieri Benth. was the shrub species with the highest IV (4.68%) in the rural site. On the other hand, *Tecoma stans* (L.) Juss. ex Kunth was the most important shrub species (7.53%) in the low urbanisation site. *Parkinsonia aculeata* L. was the most important species (14.23%) in the moderate urbanisation site. Likewise, *Caesalpinia mexicana* A. Gray was the most important species (19.63%) in the high urbanisation site (Table 1).

Clematis drummondii Torr. & A. Gray was the herbaceous species with the highest IV (4.06%) in the rural site. On the other hand, *Ruellia nudiflora* (Engelm. & A. Gray) Urb. was the most important herbaceous species (4.31%) in the low urbanisation site. *Clematis drummondii* Torr. & A. Gray also turned out to be the most important herbaceous (5.02%) in the moderate urbanization site. Likewise, *Phyla nodiflora* (L.) Greene turned out to be the most important species (7.12%) in the high urbanisation site (Table 1).

Comparisons between sites showed significant differences ($P < 0.05$) for species richness, height and coverage between all sites (Table 2). Abundance was significantly different ($P < 0.05$) between all sites, except for the comparison between the sites with moderate and high urbanisation (Table 2). All the parameters (abundance, species richness, height and coverage) decreased with increasing levels of urbanisation or pollution. In the rural site, 425.2 ± 87.7 individuals and 61.8 ± 3.4 species were registered, representing a sampling coverage of 99.9%. In the low urbanisation site, the values were reduced to 350 ± 68.5 individuals and 50.9 ± 1.6 species (coverage of 99.9%). For the moderate urbanisation site, 296.9 ± 62.4 individuals and 36.2 ± 1.6 species were registered (coverage of 99.9%), while for the high urbanisation site, 215.7 ± 35.5 individuals and 27.2 ± 0.6 species (coverage of 100%).

For ${}^{0}D$, ${}^{1}D$ and ${}^{2}D$, the rural site had the highest diversity. All comparisons between sites were significantly different (with 95% confidence intervals) (Table 2). The one-way PERMANOVA test detected significant differences in species composition between all sites ($SS_{total} = 7.05$; $SS_{within\text{-}group} = 1.25$; $F = 55.81$, $P < 0.001$). Plant communities sampled formed separate groups in the NMDS diagram (Stress = 0.11) (Fig. 3).

Table 2. Richness, abundance, height, coverage and diversity profiles along the urbanisation gradient in the MMA. Legend: ^0D = species richness expressed in units of species; ^1D = Shannon diversity expressed in units of species; ^2D = Simpson diversity expressed in units of species.

Ecological parameter	Rural	Low urbanisation	Moderate urbanisation	High urbanisation
Richness *	61.8 ± 3.4a	50.9 ± 1.6b	36.2 ± 1.6c	27.2 ± 0.6d
Abundance*	425.2 ± 87.7a	350 ± 68.5b	296.9 ± 62.4c	215.7 ± 35.5c
Height *	1.2 ± 0.1a	0.9 ± 0b	0.7 ± 0c	0.8 ± 0.1d
Coverage *	558.3 ± 103.1a	205.7 ± 31.1b	117.4 ± 22.9c	118.5 ±19.9d
^0D **	68 ± 0a	54 ± 0b	40 ± 0.2c	29 ± 0d
^1D **	61.2 ± 0.8a	47.3 ± 0.8b	35.5 ± 0.5c	25.1 ± 0.5d
^2D **	56.5 ± 1.2a	43.4 ± 1b	33.4 ± 0.7c	23.3 ± 0.7d

* Values with different letters between columns are significantly different using Kruskal-Wallis test: richness between sites, K = 36.6, DF = 3, *P* = 0·0001; abundance between sites, K = 17.5, DF = 3, *P* = 0·0001; height between sites, K = 32.5, DF = 3, *P* = 0·0001; coverage between sites, K = 31.2, DF = 3, *P* = 0·0001. ** Diversity values with different letters between columns are different, using 95% confidence intervals.

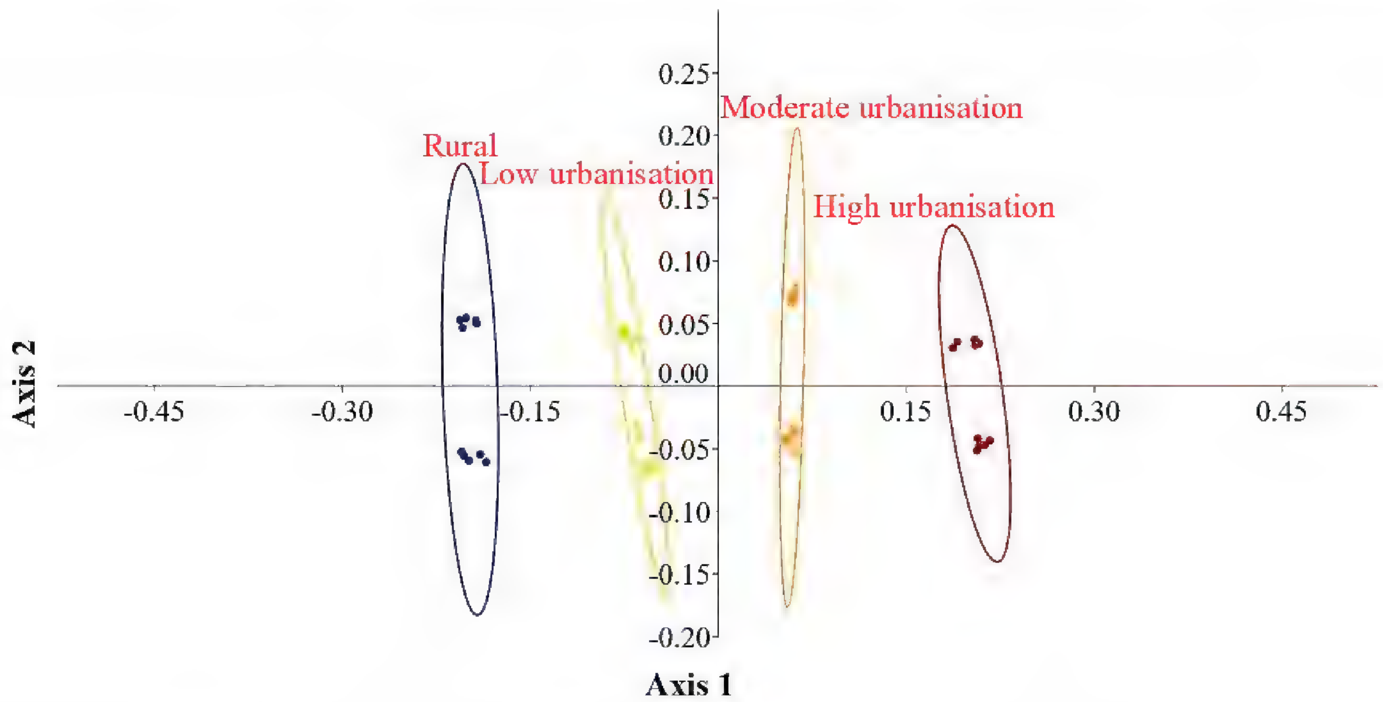

Figure 3. Non-Metric Multidimensional Scaling (NMDS) ordination of plant communities by urbanisation categories.

Plant responses to environmental variation

The MWS, T, RH, HI, DP, E, $PM_{2.5}$ and SpH were the significant environmental variables ($P < 0.05$) used in the CCA (Table 3). CCA showed significant association between the environmental variables and the plant communities along the urbanisation gradient (Total inertia = 81.3%; $P < 0.001$). The variables most related with the plant abundance in the gradient were: RH and $PM_{2.5}$ for Axis 1 (Eigenvalue = 0.441; Inertia = 56.6%). For Axis 2 (Eigenvalue = 0.193; Inertia = 24.7%), DP and HI were the most important variables. *Quercus fusiformis* Small, *Carya illinoinensis* (Wangenh.) K. Koch, *Fraxinus americana* L., *Ligustrum lucidum* W.T. Aiton, *Koelreuteria elegans* (Seem.) A.C. Sm., *Caesalpinia mexicana* A. Gray, *Punica granatum* L., *Psidium guajava* L., *Lantana camara* L., *Thymophylla pentachaeta* (DC.) Small, *Tridax coronopifolia* (Kunth) Hemsl., *Eragrostis barrelieri* Daveau, *Melinis repens* (Willd.) Zizka and *Verbena canescens* Kunth are associated with conditions of high concentration of $PM_{2.5}$, higher RH and alkaline SpH. On the other hand, *Ehretia anacua* (Terán & Berland.) I.M. Johnst., *Diospyros texana* Scheele,

Table 3. Environmental values registered along the urbanisation gradient in the MMA. Environmental variables marked (*) are significant ($p < 0.05$) according to the Monte Carlo permutation test. MWS = maximum wind speed; AWS = average wind speed; T = temperature; RH = relative humidity; HI = heat index; DP = dew point; E = evapotranspiration; SR = solar radiation; $PM_{2.5}$ = 2.5 µm particles; PM_{10} = 10 µm particles; SpH = soil pH; SM = soil moisture.

Environment variable	Rural	Low urbanisation	Moderate urbanisation	High urbanisation
MWS (Km/h) *	5.2 ± 1.5	4.2 ± 0.7	4.9 ± 2.3	15 ± 9.7
AWS (Km/h)	3 ± 1	1.9 ± 0.4	2.2 ± 0.8	1.7 ± 0.8
T (°C) *	26.4 ± 0.5	27.9 ± 1.7	26.5 ± 1.6	20.8 ± 2.7
RH (%) *	49.1 ± 6.1	62.2 ± 3.7	65.6 ± 3.8	74.6 ± 3.4
HI (°C) *	26.2 ± 1.7	32.2 ± 4.3	26.8 ± 1.2	22 ± 2.8
DP (°C) *	14.1 ± 1.9	20.9 ± 2.1	18.8 ± 1.7	16.6 ± 2
E (°C) *	18.4 ± 1.5	22.2 ± 1.3	20.4 ± 1.4	19.3 ± 1.9
SR (Klux)	9.4 ± 1.5	7.5 ± 1.5	9.1 ± 1.8	7.6 ± 3.2
$PM_{2.5}$ *	256.1 ± 101.1	410.4 ± 94.6	396.7 ± 33.4	1181.4 ± 455.6
PM_{10}	47 ± 20	71.5 ± 22.4	41.4 ± 8	69.2 ± 23.5
SpH *	7.1 ± 0.4	7.2 ± 0.6	7.3 ± 0.5	8 ± 0.7
SM (%)	11.6 ± 4.8	16.6 ± 7.2	11.7 ± 4.3	14 ± 5

Ebenopsis ebano (Berland.) Barneby & J.W. Grimes, *Prosopis glandulosa* Torr., *Sapindus saponaria* L., *Gochnatia hypoleuca* (DC.) A. Gray, *Tecoma stans* (L.) Juss. ex Kunth, *Forestiera angustifolia* Torr., *Randia obcordata* S. Watson, *Zanthoxylum fagara* (L.) Sarg., *Citharexylum berlandieri* B.L. Rob., *Verbesina persicifolia* DC., *Croton cortesianus* Kunth, *Cenchrus spinifex* Cav., *Clematis drummondii* Torr. & A. Gray, *Solanum triquetrum* Cav. and *Lantana canescens* Kunth are related to the low $PM_{2.5}$ concentration, RH and neutral SpH (Fig. 4).

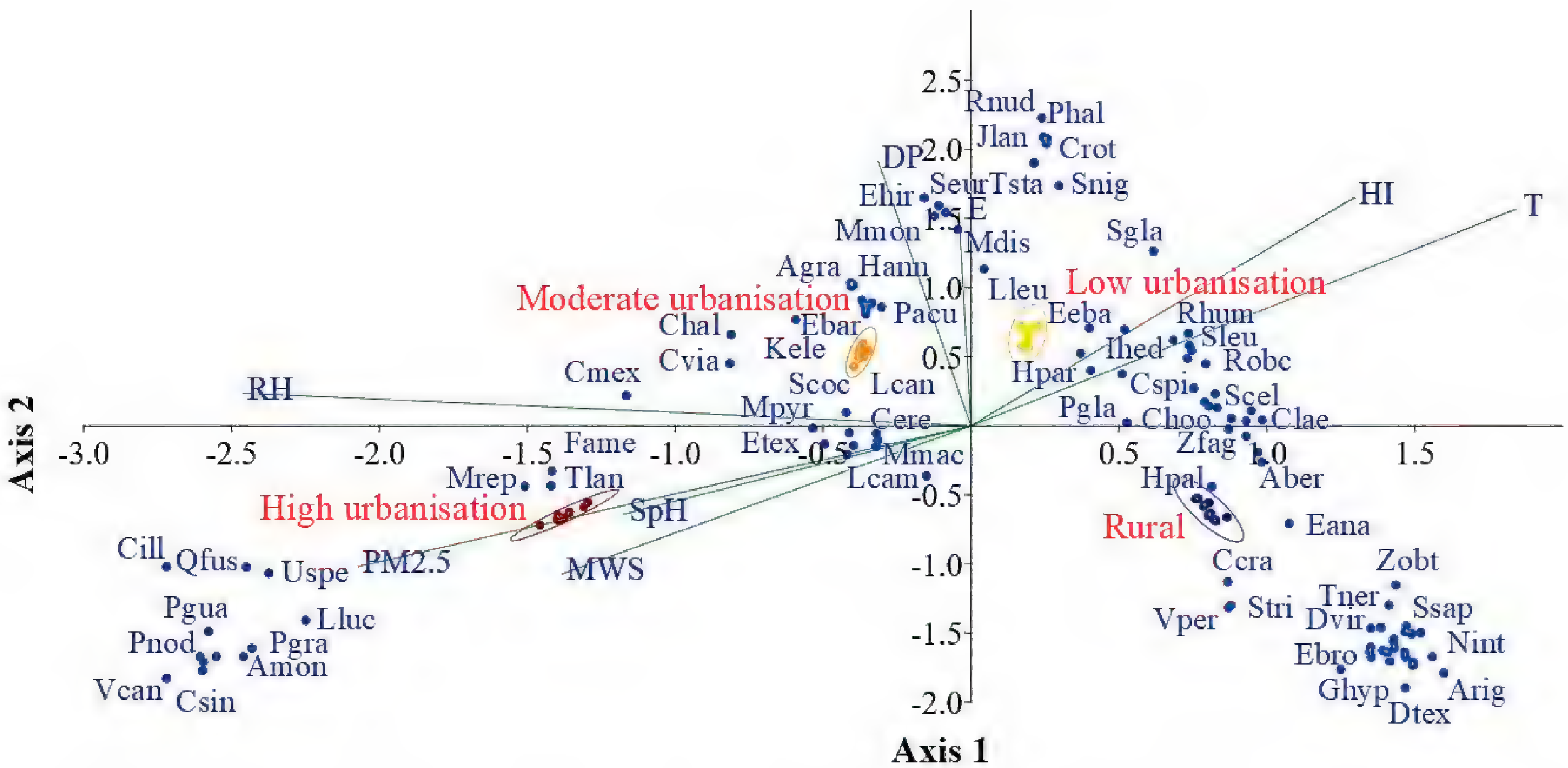

Figure 4. Canonic Correspondence Analysis (CCA) of the plant communities and significant environmental variables corresponding to the urbanisation gradient. MWS = maximum wind speed; T = temperature; RH = relative humidity; HI = heat index; DP = dew point; E = evapotranspiration; $PM_{2.5}$ = 2.5 µm particles; SpH = soil pH.

Discussion

We use the air quality records to define an urbanisation gradient in the MMA, where the height, cover, abundance, species richness and diversity were the parameters recorded in sites with different levels of urbanisation. It was found that all the parameters decreased with increasing urbanisation levels; thus, in accordance with the general disturbance hypothesis, the general tendency of plant distribution at the levels of urbanisation and pollution present in the MMA is to decrease.

It is important to note that urban gradient studies are clearly a simplification of the complex patterns produced by urbanisation, such as air pollution (Alberti et al. 2001; Hahs and McDonnell 2006; McKinney 2008). The negative effect of urbanisation on plant species richness has been related to a variety of factors ranging from the pollution and habitat degradation to introduction of alien species and others societal impacts (McKinney 2002, 2006; Hope et al. 2003; Grimm et al. 2008; Vilà et al. 2011). While the impact of urbanisation on plant richness depends upon the size of an urban area, the overall loss of habitable areas in urban zones normally results in a lower richness of plants. On the other hand, the expansion of urban areas is associated with an influx of non-native plant species that tend to counterbalance this urban effect by increasing overall plant richness (Duguay et al. 2007; Gavier-Pizarro et al. 2010; Cameron et al. 2015). These patterns were observed in the MMA region, notably, we recorded a decrease in the number of species as urbanisation levels increase and an increase in the abundance of introduced species in sites with higher urbanisation level.

The integrity of plant communities is vulnerable to intense land-use modification associated with urbanisation (Richardson et al. 2007). Significant changes in species composition along urban-rural gradients have been reported in Baltimore, Maryland (Groffman et al. 2003), Winnipeg, Manitoba (Moffatt et al. 2004) and Columbus, Georgia (Burton et al. 2005; Styers et al. 2010). Species richness and density of native plants were shown to decrease near urban areas (Porter et al. 2001; Moffatt et al. 2004), whereas invasive richness and density increased with urban development in the south-eastern United States. (Burton et al. 2005). These studies applied an urban-to-rural gradient approach to study sites located over a large geographic region from a densely populated urban landscape to a relatively unpopulated rural landscape. Our results corroborate similar studies of declining plant populations in urban-rural gradients, suggesting that habitat degradation may be a devastating threat to the persistence of certain sensitive taxa, such as plants present only in rural sites.

The replacement of local native species by alien species causes the floras of cities in different biogeographic regions to be increasingly homogeneous (i.e. beta diversity is reduced) (Kühn and Klotz 2006; Schwartz et al. 2006). However, the introduction of non-native species in urban areas can make them relatively biologically diverse at smaller scales. Our results show a clear differentiation in species composition (beta diversity) between sites on the urban-rural gradient. Hope et al. (2003) and Turner et al. (2005) demonstrate that certain anthropogenic habitats may have similar or greater alpha diversity than the more natural habitats of the region. However, our results show a greater diversity for sites without apparent urbanisation, but it decreases as urbanisation levels increase. The low diversity in such habitats may reflect a high degree of land change, thus causing significant stress to the plant community in urban areas (Pennington et al. 2010).

Certain native and alien species represent ecological indicators of different levels of urbanisation (LaPaix and Freedman 2010). *Sapindus Saponaria*, *Acacia berlandieri* and *Clematis drummondii* were the species with the highest IV value in the rural site. On the other hand, *Ungnadia speciosa*, *Caesalpinia mexicana* and *Phyla nodiflora* were the most important species in the high urbanisation site. Our species with the highest IV differ from those mentioned by Alanís-Rodríguez et al. (2015) for areas contiguous to the MMA. In contrast, Estrada-Castillón et al. (2012) report plant associations made up of species mentioned in our study, clarifying that the plant communities with the highest deterioration are associated with the areas adjacent to the metropolitan zone.

From the CCA, we identified plant species associated with urbanisation (Kremen 1992). Amongst which, invasive alien species, such as *Ligustrum lucidum*, *Koelreuteria elegans*, *Tridax coronopifolia*, *Eragrostis barrelieri* and *Melinis repens*, were found in the more urbanised sites. These species are highly tolerant to urban growth conditions and appear capable of exploiting environmental conditions associated with urbanisation (McKinney 2002). Native species, such as *Quercus fusiformis*, *Carya illinoinensis*, *Caesalpinia Mexicana*, *Lantana camara*, *Thymophylla pentachaeta* and *Verbena canescens*, were amongst the most common species to observe in urbanised sites and likely present adaptations capable of tolerating disturbance associated with urbanisation. In contrast, native species, such as *Diospyros texana*, *Sapindus saponaria*, *Gochnatia hypoleuca*, *Zanthoxylum fagara*, *Citharexylum berlandieri*, *Verbesina persicifolia*, *Solanum triquetrum* and *Lantana canescens*, were found only in the less urbanised sites. Consequently, these species are highly intolerant to process associated with urbanisation, highlighting the importance of green areas as refuges for these species. These results are consistent with the large-scale studies by Moffatt et al. (2004) and Burton and Samuelson (2008), who reported a predominance of exotic and pioneer species in more urbanised areas compared to rural areas.

The composition and structure of vegetation in peri-urban and urban areas can vary due to climate, soil conditions, ecological disturbances and human influences (Jim and Liu 2001; Jim 2002; Pedlowski et al. 2002; Escobedo et al. 2006). For this study, the conditions of RH, DP, HI and $PM_{2.5}$ were the variables that best describe the vegetation structure in the MMA. Other studies have documented these characteristics. For example, Stewart et al. (2009) in New Zealand and Godefroid and Koedam (2003) in Belgium studied different plant assemblages in urban and peri-urban temperate forests. In Latin America, Grau et al. (2008) in Tucumán, Argentina and Baumgardner et al. (2012) in Mexico City, analysed the role of the structure and composition of peri-urban forests as a function of the watershed and regional air quality, respectively. In addition, Puric-Mladenovic et al. (2000) in Canada and Christopoulou et al. (2007) in Greece, discussed the loss of peri-urban natural areas due to urbanisation.

Other anthropogenic factors of vegetation structure and composition have been found in other urban and subtropical areas of the world (Jim 2002; Grau et al. 2008). For example, people in southern China prefer green areas characterised by high tree cover and large trees (Jim and Chen 2006). Furthermore, socioeconomic and educational levels may play a role in the structure and composition of forests in Brazilian urban areas (Pedlowski et al. 2002). In Kenya, peri-urban mangroves have been affected by industrial pollution and sewage (Mohamed et al. 2009).

The approach used in this research implies a relationship between microclimatic variations and plant species at the plot level. This analysis assumes the influence of environmental variables (independent variables) on the species (Dolédec et al. 2000). However, the relationship between both factors is interdependent. That is, the structure of the vegetation and the characteristics of the plants influence the abiotic variation (Guariguata and Ostertag 2001; Renaud et al. 2010; Lienard et al. 2015; Hardwick et al. 2015) and, at the same time, the presence of certain microclimatic conditions allows the development of each plant species (Arroyo-Rodríguez et al. 2017). Therefore, the microclimate is one of the first factors to change after disturbance (Norris et al. 2012; Parr 2012; Hardwick et al. 2015).

Overall, our study analysed the effects of urbanisation on vegetation and changes in vegetation structure were detected as levels of urbanisation increased. However, studies in subtropical regions of North America show how, in addition to urbanisation, demography also affects the structure of the vegetation, mainly tree structure in built-up areas (Zhao et al. 2010; Flocks et al. 2011). Additionally, other studies in South America document the effect of socioeconomics in vegetation structure (De la Maza et al. 2002; Pedlowski et al. 2002; Escobedo et al. 2006). The ability of parks and areas of remaining native vegetation to promote biodiversity depends largely on their design and the types of management activities to which they are subjected. For example, while regionally rare native species can be found within cities, they are often associated with habitats that have not been greatly altered (Godefroid 2001; Godefroid and Koedam 2003). Such partnerships strengthen the call to protect plant communities within the urban landscape and emphasise the need for ecological knowledge to guide park design and management.

Conclusions

For the first time in north-eastern Mexico, the vegetation structure was monitored on a rural-urban gradient, where the height, cover, abundance, species richness and diversity were the parameters recorded in sites with different levels of urbanisation. It was found that all the parameters decreased with increasing urbanisation levels; thus, in accordance with the general disturbance hypothesis, the general tendency of plant distribution at the levels of urbanisation and pollution present in the MMA is to decrease. The association between environmental variables and the plant community along the urbanisation gradient was significant, the conditions of RH, DP, HI and $PM_{2.5}$ being the variables that best describe the vegetation structure in the MMA. Understanding the nature and variability of vegetation within green spaces contributes to increasing our knowledge about the distribution of the environmental services it provides and the composition of the faunal communities that depend on it. Likewise, it provides valuable information to prioritise the strategic management of the vegetation of urban green spaces so that it provides the greatest benefit for humans and biodiversity.

Acknowledgements

The first author recognises the great support of the family Cantú Bendeck for their kindness during the developing of the data collection. Bernal Herrera Fernandez gave support and recommendations during the planning of this investigation.

Additional information

Conflict of interest

The authors have declared that no competing interests exist.

Ethical statement

No ethical statement was reported.

Funding

The National Council of Sciences and Technology of Mexico (CONACyT), Ph.D scholarship No. 704911), EMJ.

Author contributions

Edmar Meléndez-Jaramillo, sampling sites selection, fieldwork, plants identification, data analysis and document writing; Laura Sánchez-Castillo, fieldwork, database compilation and document writing; Ma. Teresa de Jesús Segura Martínez, sampling sites selection, plants identification and completed document review; Uriel Jeshua Sánchez-Reyes, data analysis, results interpretation and completed document review.

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Data availability

All of the data that support the findings of this study are available in the main text.

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